Water Regulation and Population Management: Investigation into the Impacts of Dams and Foxes on Murray River Turtles and Comparison of Management Strategies

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I. Table of Contents

II. Table of Figures

III. Table of Tables

IV. List of Abbreviations

Abbreviation	Full Term
EPA	Environmental Protection Authority
IUCN	International Union for Conservation of Nature
MDB	Murray-Darling Basin
MDBA	Murray-Darling Basin Authority
MDBMC	Murray-Darling Basin Ministerial Council

V. Abstract

Freshwater turtle populations have been declining globally due to a combination of natural and anthropogenic factors, with nearly half of the world freshwater turtles threatened with extinction. Australia is high on the list of contributing countries, with one third of its native species threatened with extinction. The Murray River is an obvious example with populations of its native species *Emydura macquarii* declining by 69% and *Chelodina longicollis* declining by 91% over the past 30 years. Nest predation by the red fox and the installation of water regulators, which have affected dispersal and changed water quality, are likely main causes of these population declines

This thesis used meta-population and population viability analyses to evaluate the impacts of dams on turtle populations in areas where recruitment may be severely limited because of predation by foxes. The aims are to determine levels of recruitment required throughout a system to eliminate risks of extinction, as well as determine how the placement of dams and weirs may restrict movement and potentially increase risks of extinction. I show that turtle populations of the Murray-Darling Basin (MDB) can sustain relatively high losses of source recruitment populations, if relief populations are located between dams and impoundments. Areas of South Australia are of most concern. The scarcity of available habitats between dams, combined with high nest predation rates, increases the risk of extinction significantly. Water quality is also another significant factor that has hastened population declines and led to localised extirpation of MDB turtle populations. Management of MDB turtles centres on increasing the number of source populations for several species, increasing connectivity between populations and minimising risks for species moving terrestrially.

This study also used modelling and population viability analysis to compare the cost to benefit ratio of two management techniques; headstarting and 1080 poison baiting, to determine which management technique would require the least cost to be the most effective. Vortex modelling showed that headstarting is significantly more efficient, economically beneficial, and uniformly more successful than baiting. Headstarting cost up to 40 times cheaper than baiting yet produced the same results. Headstarting cost an average of $0.22 AUD per hectare, whilst baiting cost as much as $8.73 AUD per hectare. Furthermore, modelled headstarting success was unaffected by the proportion of waterbodies in an area, the proportion of waterbodies baited/supplemented into or the proportion of forests (areas of lower nest predation) present, and could be used uniformly throughout the river. However, baiting required more baits when a lower number of waterbodies and low proportion of forests were

present and would therefore need to be tailored to the area.

VI. Statement of Authenticity

This thesis is submitted in fulfilment of the requirements of the Master of Research course at the University of Western Sydney; School of Science and Health. The work presented in this thesis is, to the best of my knowledge, original except as acknowledged in the text. I hereby declare that I have not previously submitted this material, either in whole or in part, for a degree at this or any other institution.

[redacted]

Heather Cameron

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VII. Acknowledgements

I dedicate my thesis to my Grandfather, Jim Beaver. Thank you for always believing in me and reminding me right until the very end how proud you were of my achievements.

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1. General Introduction

1.1 Introduction

Turtles are iconic evolutionary survivors, enduring 220 million years of selective pressures which drove countless taxa to extinction (Reisz and Head 2008). However, due to a lethal combination of anthropogenic changes, freshwater turtles are now suffering a catastrophic decline on a global scale (Thompson 1983; Spencer 2000). There are currently more than 293 known species of freshwater turtles, and of these, 60 are vulnerable, 48 are endangered, 24 are critically endangered and at least 7 are extinct (Halliday and Adler 2002; Turtle Conservation Coalition 2011). That nearly half of the worlds freshwater turtles are threatened with extinction places them among the most endangered animals on earth (International Union for Conservation of Nature 2017).

Australia is not immune from these global trends, as one third of Australia's 26 native species of freshwater turtle are threatened with extinction (Wilson and Swan 2008; International Union for Conservation of Nature 2017). Threats to Australian freshwater turtles are numerous, with each life stage subject to varied threats. (Halliday and Adler 2002). Reduced recruitment due to predation of eggs by invasive mammals, increased adult mortality due to high road traffic, and limited dispersal due to dams and unfavourable habitat conditions are some of the biggest threats to Australian freshwater turtles (Thompson and Spencer 2015).

The Murray River exemplifies the extent of turtle declines in Australia. In certain sections of the river, the population size of the Murray River short neck turtle (*Emydura macquarii*) has declined by 69% and the population size of the eastern long neck turtle (*Chelodina longicollis*) has declined by 91% over the past 30 years (Chessman 2011). A slight decrease in the population size of the broad shelled turtle (*Chelodina expansa*) was also observed, however it was not large enough to be concerned (Chessman 2011).

I aim to study the impact of anthropogenic disturbances and fox predation on freshwater turtle species within the Murray River, and explore solutions or methods to reduce the damage caused by these factors. Headstarting, i.e. the captive breeding and hatching eggs of a species before supplementing them into wild populations (Heppell et al. 1996; Mitrus 2005), is a possible method to reduce the effects of juvenile and adult mortality and reduced dispersal. Lake Bonney-Barmera, South Australia, a freshwater lake fed and drained by the Murry River, is a good indicator of the level of population damage as there has been an observed decline in the population size of the three-sympatric species of Murray River turtle (*E. macquarii*, *C. expansa* and *C. longicollis*) which is hypothesised to continue to a detrimental end (Thompson

1983). Therefore, this location can be used to study the effectiveness of headstarting on stabilizing or increasing juvenile recruitment and population numbers.

1.2 Specific aims and hypothesis

In this thesis, I will test the following questions:

1. Will the combination of reduced juvenile recruitment from high nest predation and dams that restrict movement and dispersal increase the probability of extinction of Murray River turtles
2. Are standard fox management techniques eg (lethal baiting) or headstarting more economically viable.

1.3 Threats to turtles

1.3.1 Recruitment

All of Australia's freshwater turtles are members of the family Chelidae, which lay on average 1-2 clutches of 10-25 brittle shelled eggs per year (Cronin 2001). Therefore, at maximum reproductive potential each female can lay up to 75 eggs per year. In natural conditions, a large portion of these hatchlings/juveniles will not survive to reproductive age, as nests are destroyed by native predators including goannas, water rats, and birds such as crows and magpies (Cronin 2001; Romanowski 2013). This high level of juvenile mortality is counteracted by these turtle's high fecundity, therefore a few surviving females from each season is sufficient to maintain population sizes and structures (Thompson 1983; Spencer 2000).

This ecological balance was challenged by the 1871 introduction of the European red fox (*vulpes vulpes*) (Saunders 1995). Foxes are highly adaptive, opportunistic predators exhibiting surplus killing behaviour, whereby they kill prey at a rate beyond their immediate nutritional requirements (Saunders 1995; Short et al. 2002). The fox is one of Australia's worst vertebrate pests, and inflicts widespread damage to livestock and native populations (Saunders 1995; Spencer 2000). Fox predation may result in 4-30% of agricultural lamb loss in Australia. Foxes also commonly prey upon goat kids, poultry, and other livestock animals, resulting in upwards of $2 million of economic loss for Australian agriculture annually (Lugton 1993; Greentree et al. 2000; Gong et al. 2009). Furthermore, the introduction of foxes has played a major role in the decline of ground-nesting birds and small mammals such as the Mallee fowl

(*Leipoa ocellata*), little tern (*Sterna albifrons*), long-footed potoroo (*Potorous longipes*), southern brown bandicoot (*Isoodon obesulus*), Hastings River mouse (*Pseudomys oralis*), yellow-footed rock-wallaby (*Petrogale xanthorpus*), mountain pygmy-possum (*Burramys parvus*), brush-tailed rock-wallaby (*Petrogale penicillata*), and broad-toothed rat (*Mastacomys fuscus*; Burbidge and McKenzie 1989; Department of Sustainability, Environment, Water, Population and Communities 2010).

The impact of foxes on Australian reptiles is relatively unknown, however foxes, like native predators, excavate turtle nests and break open the eggs to consume the contents (Thompson 1983; Spencer 2000). A few foxes can destroy and consume a much larger number of eggs in a comparatively shorter time than native predators, therefore their destructive capabilities are higher than those of native predators (Burbidge and McKenzie 1989). Australian freshwater turtles suffering heavy fox predation show a lack of juvenile recruitment (Thompson 1983). Murray River turtles are suffering considerably due to fox predation, with 96% of nests being taken by predators, and 93% of this predation being by foxes (Thompson 1983). Juvenile recruitment rate was as low as 4%, which is not high enough to allow adequate breeding rates and maintain the population (Thompson 1983). Australian freshwater turtles may be suffering from an extinction debt, whereby the lack of juvenile recruitment is masked by long life spans and large populations of adult turtles, but leads to an eventual population collapse when aging adults eventually die off (Browne and Hecnar 2007).

1.3.2 Adult mortality

Adult turtles, particularly females, are also vulnerable to fox predation. Their hard carapaces protect them from most natural predators in the water, however when females venture onto land to nest they are left exposed and vulnerable (Spencer 2000). Increasing road traffic is also increasing mortality rate of adult turtles (Doroff and Keith 1990; Brooks et al. 1991). During the nesting season, females often venture from the water in search of nesting grounds, crossing busy roads in the process (Gibbs and Shriver 2002; Gibbs and Steen 2005). Although millions of years has allowed turtles to develop a hard carapace that is resistant to most predators, these shells are not adapted to withstanding the force of a car, and easily crack under the pressure.

Due to the high mortality rate of juvenile turtles, population survival of freshwater turtles relies on high survival rates of adult turtles to facilitate high reproduction rates. Increasing adult female mortality due to road deaths is a threat as it reduces the breeding potential of the population (Gibbs and Steen 2005). Female freshwater turtles can take up to

15 years to reach sexual maturity (Georges et al. 2003), therefore if a large portion of the breeding population is wiped out by high road mortality, it will take a considerable amount of time for juveniles to reach reproductive age and reproduce enough to balance out the population damage (assuming some recruitment is occurring despite high fox predation of juveniles).

Although the effects of road mortality on freshwater turtle populations within Australia have not been quantified, studies conducted on American freshwater turtles have indicated that annual road based mortality in high traffic areas can be >5% of a population (Gibbs and Shriver 2002). Long term studies of freshwater turtle mortality have indicated that as little as 2–3% additive annual mortality is the most turtle species can withstand and still maintain positive population growth (Doroff and Keith 1990; Brooks et al. 1991). Furthermore, a higher mortality for adult females may result in unequal sex ratios within populations. Gibbs and Steen (2005) synthesised published estimates of sex ratios within various freshwater turtle populations from 1928-2003 and analysed these results against changes to road networks within these areas. They suggested that the proportion of males compared to females in populations residing in areas with high road densities had increased considerably, and that this male bias was synchronised with the expansion of the road network. These results are mirrored in many similar studies, such as Aresco (2005), Marchand and Litvaitis (2004) and Steen et al. (2006), all of which concluding that high road mortality may reduce the proportion of females within populations, resulting in male-biased sex ratios.

Adult mortality is also increasing due to growing frequency of boat collisions in recreational waterways. The effects of boat mortality have not been quantified within Australia, however in Ontario, Canada an analysis of 312 freshwater northern map turtles (*Graptemys geographica*) found that just over 20% had injuries consistent with boat propeller collisions, and females had a much higher boat injury rate (28.6%) compared to males (12.8%) (Bennett and Litzgus 2014). Additionally, within Australia adult freshwater turtles have suffered from mass mortality due to the recent 'millennium drought' (Roe and Georges 2010; Chessman 2011). During this drought, historically permanent wetlands dried up, resulting in many of the adult turtles perishing from inability to escape to other water bodies. This drought also triggered a rise of salinity in the fringing wetlands and lower lakes of South Australia, causing excessive growth of estuarine tubeworms (*Ficopomatus enigmaticus*) on the carapaces of turtles which in some instances encased 100% of the carapace and inhibited limb movement, resulting in high mortality (Bower et al. 2012). Under current predictions of climate change, droughts comparable to the millennium drought may become commonplace (Roe and Georges 2010), boding ill for adult mortality rates.

1.3.3 Dispersal

Many of Australia’s freshwater turtles, such as the Murray River short-neck turtle (*E. macquarii*) and the broad shelled turtle (*C. expansa*) rarely migrate terrestrially, and only disperse via swimming (Baggiano 2012). Over the past 150 years, hundreds of dams and weirs have been constructed throughout Australia, and the impacts that these water impoundments may have on freshwater turtle populations is unknown. Dams create barriers for upstream and downstream movement, which become more prominent during periods of low flow (Baggiano 2012). Dam barriers may greatly reduce or eliminate dispersal of individuals between fragmented populations, particularly for primarily aquatic species such as *E. macquarii* and *C. expansa* (Baggiano 2012).

Dispersal among populations is one of the key processes ensuring survival of a species in the fragmented ecosystems created by the creation of dams and weirs (Hanski 1998; Clobert et al. 2004). Reduced dispersal among fragmented populations or ‘metapopulations’ reduces or eliminates gene flow among these populations (Whitlock and McCauley 1999). Reduced gene flow can be dangerous for isolated populations as it increases the negative effects of mutation (the formation and accumulation of new alleles), genetic drift (the loss of alleles through generations resulting from reproduction), and selection (the preferential survival or elimination of specific alleles/genes through generations; Slatkin 1987; Neuhauser 2007). Metapopulations which receive little to no genetic exchange from other populations have a much higher chance of mutating deleterious alleles, i.e. alleles that cause premature death, serious health problems or a highly compromised reproductive capacity (Hanski 1998). A high proportion of deleterious alleles within generations can result in excessive adult and juvenile mortality as well as widespread reproductive failure, which has the potential to wipe out entire metapopulations (Hanski 1998).

1.3.4 Habitat

Anthropogenic habitat changes pose an additional threat to Australian turtles, particularly within the Murray River. Since European settlement, the natural flow regimes of the Murray River and the wider Murray-Darling basin have been altered by dams, weirs, lakes and barrages to produce a more reliable water supply for the growing agricultural industry along the river (MDBA 2011; Docker and Robinson 2014). This environmental manipulation and exploitation however has reduced the amount of water in the system, disrupted natural flooding patterns and compromised the overall health of the river and its ecosystems (MDBA 2011).

The extensive environmental degradation within the Murray-Darling system includes salinization, high turbidity, high sediment loads, and alteration of flood plains (MDBMC 2002; MDBA 2010; MDBA 2011). Salinization, as stated previously, poses a great risk to turtles as it triggers excessive and sometimes lethal growth of estuarine tube worms on the carapaces of turtles and on other macro invertebrates (Bower et al. 2012). It is also possible that turtles exposed to high salinity water may suffer dehydration (Seidel 1975; Kinneary 1993) or that they may encounter difficulty transferring water from their interstitial fluid to the external water due to the increased intake of ions raising the osmolarity of the turtle (Lee et al. 2006).

Following anthropogenic development, sedimentation, and turbidity are the most significant causes of freshwater degradation (Henley et al. 2000). There have been few studies on the effects of increased suspended sedimentation and turbidity on the ecology of freshwater turtles, however there are indications that they could be detrimentally affected by it (Tucker 1999; Grosse et al. 2010). Increased sediment reduces the dissolved oxygen in the water column (Herbert and Merkens 1961). Many of Australia's endemic freshwater turtles, including genera *Elseya*, *Rheodytes*, *Elusor*, and *Myuchelys* exhibit bimodal respiration, using cloacal respiration as well as normal respiration to minimise surfacing frequency and extend dive duration (Legler and Georges 1993; King and Heatwole 1994). High suspended sediment may be detrimental to bimodally respiring turtles as it may impair cloacal respiration and thus reduce dive duration (Tucker 1999).

Regarding turbidity, fish, which are often visual predators relying on water clarity for foraging efficiency, have been shown to have a reduced feeding efficiency in high turbidity environments due to prey detection being obscured (Cezilly 1992). Whether high turbidity has similar effects on freshwater turtles has not been addressed, however freshwater turtles are known to have acute vision and rely heavily on visual detection of prey (Sexton 1959; Parmenter and Avery 1990; Ernst and Lovich 2009) therefore it is possible that these species may also suffer from reduced feeding efficiency. Turbidity can also reduce total primary productivity in freshwater systems, which has potential to reduce the food available for all species as a result of trophic cascades (Personal communication, J. Van Dyke). Furthermore, historically wetlands along the Murray have experienced periods of drying and wetting coinciding with natural flooding events, however alterations to the natural flow of the river have resulted in important flooding events being lost, resulting in some wetlands permanently drying or being permanently inundated and reducing viable and accessible turtle habitat (Docker and Robinson 2014).

1.4 Conservation methods

1.4.1 Headstarting

Headstarting is a conservation tool that is gathering increasing attention as a solution to the catastrophic population decline of freshwater turtles. The method involves the artificial incubation, hatching and raising of neonate animals in captivity, before release of juveniles into the wild (Mitrus 2005). Headstarting has potential to be successful for Australian freshwater turtles, as it protects the most vulnerable life stages; eggs and hatchlings. By incubating and hatching turtle eggs in captivity, they would avoid nest predation by foxes, pigs and native animals, theoretically ensuring a higher proportion of juvenile recruitment into the population. This increase in juvenile recruitment may also compensate for the detrimental effects of adult mortality (Heppell et al. 1996; Mitrus 2005). The controlled release of juveniles into the wild also allows for population dispersal rates to be factored, therefore if some metapopulations are suffering from lack of immigration, juveniles can be released into these populations to stabilize them.

Headstarting programs have been successfully implemented globally for many species of marine turtle, including green sea turtles (*Chelonia mydas*; Bell et al. 2005), and various species of Amazonian turtles (Alho 1985). The technique is also becoming increasingly common for other reptile species, being trialled for Caribbean rock iguanas (*Cyclura* sp.) (Alberts 2007), West Indian iguanas (Alberts et al. 2004), the plains gartersnake (*Thamnophis radix*; King and Stanford 2006) and green iguanas (*Iguana iguana*) (Escobar et al. 2010).

Perhaps the greatest indication that headstarting is a valuable turtle conservation method is that the 'Turtle Survival Alliance', the world's most prominent turtle conservation organisation, is currently operating headstarting programs for at least 11 turtle species. These species include two land tortoises; the ploughshare tortoise (*Astrochelys yniphora*) and the Burmese star tortoise (*Geochelone platynotan*), and 9 species of freshwater turtle; the northern river terrapin (*Batagur baska*), southern river terrapin (*Batagur affinis*), painted terrapin (*Batagur borneoensis*), red-crowned river terrapin (*Batagur kachuga*), Burmese roofed turtle (*Batagur trivittata*), Indian narrow-headed softshell turtle (*Chitra indica*), Asian narrow-headed softshell turtle (*Chitra chitra*), alligator snapping turtle (*Macrochelys temminckii*) and Magdalena river turtle (*Podocnemis lewyana*) (Burke 2015).

Although the Turtle Survival Alliance is undertaking many headstarting studies on freshwater turtles, few headstarting studies have lasted long enough to report population benefits, therefore the efficiency and success of headstarting for freshwater turtles is relatively unknown. There have however been two studies which suggest that headstarting may be effective for freshwater species. Townsend et al (2005) conducted a headstarting study in Ecuador on the freshwater arrau turtle (*Podocnemis expansa*) and the yellow spotted river turtle (*Podocnemis unifilis*) over 7 years which noted a substantial increase in population numbers at the end of the study. Similarly, a headstarting study by Mitrus (2005) in Poland on the European pond turtle (*Emys orbicularis*) concluded with a 42% survival rate of head-started hatchlings compared to a 7.5% rate for wild hatchlings.

There have however been numerous papers critical of headstarting, with three main criticisms commonly explored in these papers. Firstly, the life history of freshwater turtles involves high adult survivorship coupled with low hatchling and juvenile survivorship, with survival relying on exceptionally long lifespans and multiple reproductive opportunities for adult females. Population models (e.g., Congdon et al. 1993; Heppell 1998; Heppell and Crowder 1998) have shown that population persistence is more sensitive to survivorship of adults than that of younger life stages. Therefore, due to generally limited conservation resources and the fact that headstarting focuses on the protection of young life stages, it would be more valuable to protect adults than to head-start eggs and hatchlings, as turtle populations can survive many years with limited recruitment but are quickly eradicated if adult survivorship is low (Heppell 1998; Heppell and Crowder 1998).

Secondly, headstarting usually does not address the most prevalent conservation issues for freshwater turtles (e.g. adult mortality or habitat quality), and therefore can be a waste of resources or a detrimental distraction from effective conservation techniques (Klemens, 2000; Seigel and Dodd 2000). For example, if there are habitat issues which are affecting turtle survivorship, releasing head-started hatchlings will not address conservation concerns. Thirdly, headstarting programs may fail or be detrimental to the population if head-started hatchlings behave differently to wild hatchlings or are diseased (Klemens 2000; Seigel and Dodd 2000). And finally, existing studies may be too short-term as there is no evidence that headstarted turtles survive to adulthood and reproduce. For headstarting to have real benefit, these results must be observed, and it is possible that they may not survive to adulthood or they survive to adulthood but fail to reproduce (Klemens 2000; Seigel and Dodd 2000).

1.4.2 Reduction of foxes

In all Australian states, a combination of invasive pest control methods including poison baiting, shooting, and den fumigation have been implemented in response to the high level of fox predation on native animals (Department of Primary Industries 2015; Agriculture Victoria 2017a). Sodium fluoroacetate, also known as poison '1080', is often used for poison baiting (Saunders and McLeod 2007; Department of Primary Industries 2015). 1080 is a colourless, odourless, and tasteless compound that occurs naturally in many poisonous plants in Australia (Department of Agriculture and Fisheries 2016; Agriculture Victoria 2017b). The poison is highly toxic to mammals and works by interrupting the animal's cellular respiration (krebs) cycle, blocking the release of energy and resulting in death by respiratory or heart failure (Agriculture Victoria 2017b). Generally, there are three forms of the bait used throughout Australia; shelf stable baits, perishable baits, and canid pest injectors. Shelf stable baits are dried meat-based baits which have a shelf life of up to two months (Agriculture Victoria 2017a; Agriculture Victoria 2017b), where-as perishable baits have a shelf life of up to three days, and are made up of fresh meat (generally liver for foxes) injected with an aqueous 1080 solution (Agriculture Victoria 2017a; Agriculture Victoria 2017b). Canid pest injectors are spring loaded devices which are buried in the ground with a lure attached. The animal pulls on the lure, triggering a spring-loaded plunger that punctures a capsule containing 3mg of 1080 and propels it directly into the animals' mouth (Department of Primary Industries 2015; Agriculture Victoria 2017b).

Baiting with 1080 is regarded as environmentally safe, as it is not a persistent compound and can be metabolized by biological systems (United States Environmental Protection Agency 1995; Eason 2002). In sterile and cold environments 1080 may degrade more slowly, with the potential to leach into water, however it is considered non-toxic to aquatic invertebrates and only slightly toxic to freshwater fish (Eason 2002). Baiting with 1080 however does pose the risk of harming non-target species, particularly domestic dogs, birds, agricultural stock, and native mammals (Office of Environment and Heritage 2011). This risk can be reduced by ensuring the bait is properly buried, restraining domestic dogs and stock from accessing the baited area and avoiding baiting during native mammal breeding seasons (Office of Environment and Heritage 2011; Agriculture Victoria 2017a).

After poison baiting, shooting is the second most popular method for fox control (Department of Primary Industries 2015). Shooting is generally conducted on a small scale by landholders and local hunting groups, and is regarded as the most target-specific and humane form of fox control (Agriculture Victoria 2017a). For individuals and small groups, spotlight shooting in late summer and early autumn is the most effective to eliminate many foxes, as adult foxes are more active at night and young cubs can be attracted with a fox whistle easily at this time. However, the number of fatalities drops after a few nights due to the danger recognition response of the foxes (Agriculture Victoria 2017b). For larger fox hunting groups, daylight drives or battues can be more effective, although this technique is highly labour intensive (Department of Primary Industries 2015; Agriculture Victoria 2017a). Shooting must be conducted in conjunction with alternative forms of fox control however, as this method alone will not achieve long term fox control (Agriculture Victoria 2017b). Foxes become educated on potential dangers quickly, and after a short period of shooting foxes become wary and less visible, resulting in difficulty estimating fox numbers and eliminating the remaining population (Agriculture Victoria 2017a; Agriculture Victoria 2017b).

Den fumigation is a less common but highly effective means of controlling fox populations (Department of Primary Industries and Regional Development 2016; Agriculture Victoria 2017a). Vixens generally excavate dens from April to May, and cubs are born from August to September (Department of Sustainability, Environment, Water, Population and Communities 2010). Vixens also habitually re-use dens, and are confined with their new born cubs to the den from August to October, therefore it is this period when fumigation is the most effective (Agriculture Victoria 2017a). A carbon monoxide fumigant is the most commonly used for this method, and if used during the confinement period can successfully kill both the vixen and cubs (Department of Primary Industries and Regional Development 2016). This technique is labour intensive, and most effective if the dens are destroyed post fumigation, and sweeps of commonly habituated areas are conducted annual from May to June and August to September to destroy and fumigate dens before and after cub birth (Department of Primary Industries and Regional Development 2016).

1.5 Study Site: The Murray River

This study will take place within the Murray River. The Murray is 2530km long and spans three states (New South Wales, Victoria and South Australia; Department of Environment, Water and Natural Resources 2014). The Murray is fed by several rivers, including the Darling and the Murrumbidgee.,The Murray-Darling Basin serves as Australia’s largest drainage area and one of the largest in the world, draining most of southern Queensland, New South Wales and inland Victoria (Department of Environment, Water and Natural Resources 2014; MDBA 2014).

The Murray River and the wider Murray-Darling basin have served as an important resource for Aboriginal people for millennia, with countless aboriginal groups utilising its resources (MDBA 2014). In the 1880’s, the Murray became a vital hub for agriculture, with paddle-steamers used to carry wheat, wool and other goods up and down the river system (Department of Environment, Water and Natural Resources 2014). True exploitation of the river began with the 1887 introduction of an irrigation system, and further with the formation of the Murray River commission (now the Murray-Darling Basin authority) in 1918 (Guest 2016). The commission led the construction of 4 major dams, 16 storage weirs and 15 locks along the Murray, with the intention to reduce the effects of drought and flood (MDBA 2012). Although only 36% of natural flow now reaches the mouth of the Murray, the water management measures have ensured the maintenance of a constant flow of water in the river system (MDBA 2014) however this has come at a price for the ecology and environment of the Murray.

Lake Bonney-Barmera, South Australia will be the location where I conduct the field research for my thesis (discussed in conclusions and recommendations). Lake Bonney-Barmera is a freshwater lake of the Riverland region of South Australia, with the town of Barmera located on its shores. The lake is fed and drained by the Murray River, and has a surface area of approximately 16.2km^2(EPA 2003). The lake is a hub of environmental life, with its banks home to black swans, pelicans, emus, bearded dragons and western grey kangaroos, and its waters plentiful with Murray cod, bream, perch, redfin, carp and turtles (EPA 2003).

1.6 Murray River Turtles

Freshwater turtles, in particular the three-sympatric species of Murray River turtle, are the study species for my thesis. Freshwater turtles serve an important ecological role, controlling the levels of algae, microbes, plants and animals within their environments (Chessman 1978), and often representing a large biomass and significantly impacting nutrient cycling and energy flow within freshwater systems (Thompson 1993; Souza and Abe 2000). Up to 230,000 tonnes of turtles inhabit the Murray River in Victoria, which are believed to consume from 130 to 430 tonnes of carrion each day, or 180,000 tonnes of carrion per season (Thompson 1993). Depending on species, carrion consumption can represent up to 80% of a freshwater turtle’s diet, therefore some turtles have the capacity to have significant effects on nutrient cycling and energy flow of the Murray River (Chessman 1986; Georges, Norris and

Wensing 1986). Freshwater turtles are also important seed dispersers for terrestrial riparian plants (Legler 1976; Kennett and Tory 1996)

There are three species of turtle which inhabit the Murray River: The Murray River short-neck turtle (*E. macquarii*), broad-shelled turtle (*C. expansa*) and eastern long-necked turtle (*C. longicollis*). All three species are from the Chelidae family and are pleurodiran (side-necked; Chessman 1988; Cogger 1975). The reproductive strategies of these species vary. *C. longicollis* and *E. macquarii* nest in late spring to early summer, from September to December, whereas *C. expansa* nests mainly in Autumn from March to May, although substantial rains can alter the nesting season (Cann 1998). Nests of Chelidae species are generally laid within a few metres from the water's edge, however due to environmental conditions nest distance can be up to 1km from the water in some cases (Goode and Russell 1968; Cann 1998). Typically, 10-15 eggs are laid each nesting season, although depending on individual health and ecosystem quality this can vary from 5 to 30 eggs per season. Incubation periods range depending on species, from as little as 65-85 days in *E. macquarii*, and 120-180 days in *C. longicollis* to 200-650 days in *C. expansa* (Goode and Russell 1968; Cann 1998). Sex determination is genetic in *C. longicollis* and *E. macquarii*, but unknown for *C. expansa* (Ezaz et al. 2006; Georges 1988)

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2. Modelling the impacts of variable predation and water regulation on Murray River turtles

2.1 Abstract

Freshwater turtles in the Murray River are declining rapidly, with populations of *Emydura macquarii* and *Chelodina longicollis* dropping by 69% and 91%, respectively, over the past 30 years. Nest predation by the red fox and the installation of water regulators, which have affected dispersal and changed water quality, are likely main causes of these population declines. This study uses meta-population and population viability analyses to evaluate the impacts of dams on turtle populations in areas where recruitment may be severely limited because of predation by foxes. The aims are to determine levels of recruitment required throughout a system to eliminate risks of extinction, as well as determine how the placement of dams and weirs may restrict movement and potentially increases risks of extinction. I show that turtle populations of the Murray-Darling Basin (MDB) can sustain relatively high losses of source recruitment populations, if relief wetlands are located between dams and impoundments. Areas of South Australia are of most concern. The number of available habitats between dams, combined with high nest predation rates, increases the risk of extinction significantly. Management of MDB turtles centres on increasing the number of source populations for several species, increasing connectivity between populations and minimising risks for species moving terrestrially.

2.2 Introduction

The world's freshwater turtles are being driven to extinction by a combination of natural and anthropogenic factors. Climate change (McCallum et al. 2009), habitat loss (Turtle Conservation Coalition 2011), poaching (Turtle Conservation Coalition 2011), introduced predators (Spencer 2000), and disease (Dodd 1998) are reducing populations in the wild at an alarming rate, with 191 of the 263 known species of freshwater turtle in danger of extinction (Turtle Conservation Coalition 2011). Australia is prominent on the IUCN's red list of endangered species, and as of 2015, ten of its 23 native species of freshwater turtle were listed as either vulnerable, endangered, or critically endangered (International Union for Conservation of Nature 2014).

In Australia's largest river system, the Murray-Darling Catchment, populations of the Murray River short-neck turtle (*Emydura macquarii*) have declined by 69%, while populations of the eastern long necked turtle (*Chelodina longicollis*) have declined by 91% in the past 30 years (Chessman 2011). The introduced red fox (*Vulpes vulpes*) has contributed significantly to the declines in populations of freshwater turtles and is also implicated in the extinction of 30-40 native Australian species (Saunders et al. 1995). Foxes are responsible for 93% of nest predation in the lower regions (Thompson 1983) and 90% in the upper regions (Spencer 2002) of the Murray River. Although population declines have been observed (Chessman 2011), the effect of fox predation on the long-term health of Murray River turtle populations may be masked by their longevity. Murray River turtles exhibit a type III survivorship curve (Iverson 1991; Spencer 2000). Mortality is greatest in the egg, hatchling, and juvenile life stages, while adult survival is relatively high (Shine and Iverson 1995). Type III populations mainly consist of adults, with low juvenile recruitment. Freshwater turtles also exhibit long lifespans. *Chelodina longicollis* survives in captivity for at least 36 years (Goode 1967), and in the wild for potentially upwards of 100 years (Parmenter 1985). Long lifespans mean that a continual lack of juvenile recruitment due to fox predation can take generations to affect population densities. If recruitment never improves, the adult population will slowly disappear via old-age attrition (Thompson 1993).

Dams may also impact freshwater turtles in the Murray River, through changes in habitat and water quality, reduced environmental flows, and reduced dispersal capabilities. Within the Murray River, there are four major dams, 16 storage weirs, and 15 locks (MDBA 2012), and in the wider Murray-Darling river system there are over 100 water impoundments (Walker 1979). Dams, weirs, and locks have been shown to have a detrimental impact on flora and fauna, particularly macroinvertebrates and fish (Bell et al. 1980; Chessman et al. 1987; Doeg et al. 1987; Marchant 1989; Walker and Thoms 1993). Two native species, the golden perch (*Pectroplites ambiguous*) and freshwater catfish (*Tandanus tandanus*), have disappeared since the construction of the Hume weir, and several introduced species such as redfin (*Perca fluviatilis*), tench (*Tinca tinca*) and golden carp (*Carassius auratus*) are becoming increasingly dominant (Walker et al. 1978).

Many management practices, such as fish ladders and carp screens, have been established to reduce the effect of dams and other anthropogenic factors on Murray River fish species, however little consideration has been afforded to freshwater turtles. Fish ladders provide connectivity between fragmented habitats through dams and reduce predation on groups of fish that generally congregate at the bottom of dams (Reynolds 1983; Agostinho et al. 2007). Carp screens control the numbers of larger carp entering ecologically important wetlands by excluding larger carp from passing through (Hillyard et al. 2010). Both fish ladders and carp screens however block the movement of freshwater turtles and can cause turtles to drown (Calles and Greenberg 2009).

Water regulation in the Murray has not only reduced connectivity along the river but also laterally into the floodplains (Walker 1985). Flows in the river were once highly variable, with large floods promoting high levels of recruitment of flora and fauna in the floodplain and riverine communities, while seasonal floods maintained lower levels of recruitment (Walker and Thoms 1993). River regulation has also limited exchanges of water between the floodplain and the river, causing typically floodplain species to invade areas formerly occupied by species adapted to life in flowing water (Walker 1985). Several riverine species, particularly fish, have also suffered a decline in abundance and range (Walker 1985). There has however been minimal investigation on the impacts of these water impoundments on freshwater turtles.

Two of the Murray River's three native species of turtle (*E. macquarii* and *C. expansa*) rarely migrate terrestrially, and primarily disperse via swimming (Baggiano 2012). Impoundments create barriers for aquatic movement, potentially reducing or eliminating dispersal of *E. macquarii* and *C. expansa* (Baggiano 2012). *Chelodina longicollis* disperses terrestrially and aquatically, but changes to flooding regimes may force them to migrate terrestrially more often (Baggiano 2012). Dispersal between the fragmented ecosystems created by the construction of dams and weirs is vital to ensure the survival of the species (Hanski 1998; Clobert et al. 2004). Reduced dispersal between metapopulations (groups of populations that are separated by space but consist of the same species) limits breeding potential, stops the replenishment of populations which may have suffered higher mortality than neighbouring populations, particularly due to fox predation, and reduces gene flow (Hanski 1998; Clobert et al. 1999; Ims and Rousset 2004). The combination of fragmented habitats that limit dispersal and high nest predation may be the major drivers of population declines within the Murray River (Chessman 2011).

Terrestrial habitats near the Murray River include three primary types; agriculture, urban, and natural (Brown and Stephenson 1991; Sinclair 2001). Most nest predation studies have focused on agricultural or urban areas, and report high nest predation rates as foxes are abundant in these areas (Thompson 1983; Spencer 2000). Forests may potentially have a lower predation rate due to habitat heterogeneity (R. Spencer, unpublished data). Forested areas provide cover and protection for nests, and could blur the olfactory and visual cues that foxes use to find nests. Agricultural areas with monocultures of grasses and crops offer comparatively limited protection. Thus, broad land management relative to the location of dams may have significant implications for turtle recruitment throughout the system.

This study uses meta-population and population viability analyses to evaluate the impacts of dams on turtle populations in areas where recruitment may be severely limited because of predation by foxes. The aims are to estimate levels of recruitment required to eliminate risks of extinction at a landscape level, and model how the placement of dams and weirs restrict dispersal and increase risk of extinction.

2.3 Materials and Methods

The Murray River is Australia's longest river. The river is 2,508 km long, originating near Mount Kosciuszko in the Australian Alps, forming the border between New South Wales and Victoria, flowing northwest into South Australia, and reaching the ocean at Lake Alexandrina (Sinclair 2001; Young 2001).

I surveyed the number of water bodies, as well as the proportions of terrestrial environment type (forest, agriculture, and urban) from the Hume weir to the river mouth to create a model of the number of turtle habitats in the Murray River. I used Google Earth and Google Maps to conduct this survey, and I divided the river into sections of 10 x 10 km areas (Fig. 2.1). I defined waterbodies as follows:

"Pond": a small wetland isolated from river,
"Lake": large reservoir or natural lake,
"Lagoon": lentic wetland with a narrow (<10 m) connection to main river,
"Backwater": large lentic wetland with a broad (>10 m) connection to main river,
"Tributary: slow-moving main-branch tributary to Murray,
"Anabranch": a stream that branches off the main channel before re-attaching.

The total and average number of each type of water body between each dam, weir, and lock was calculated (Fig. 2.1). These values were used to create metapopulation models to assess population viability.

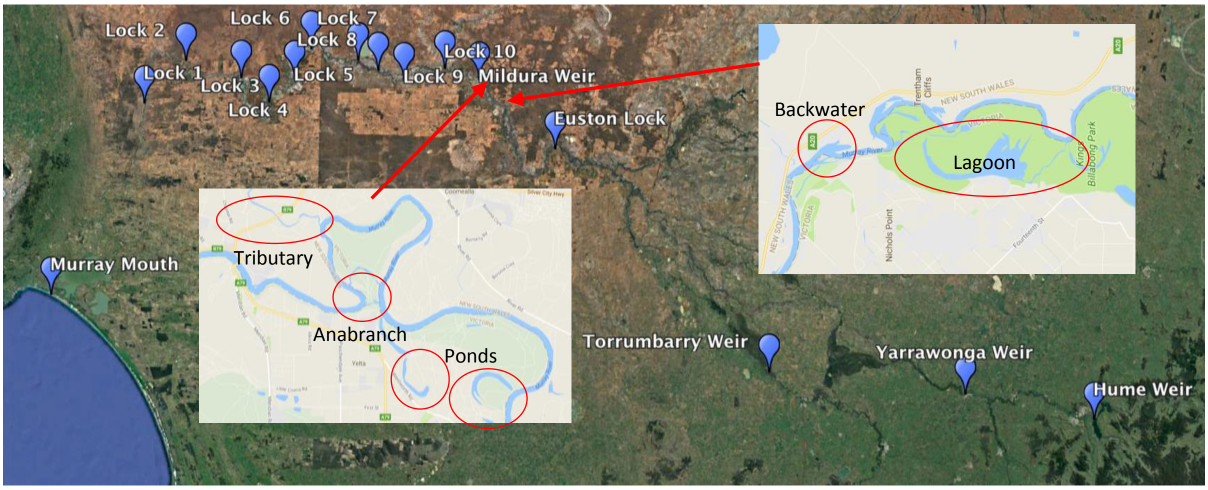


Fig. 2.1. Location of dams, locks, and weirs along the Murray Rivers, as well as, map sections used to classify the number and types of habitats.

VORTEX (Version 10.0) was used to model extinction probabilities in the Murray River for freshwater turtles, following previous studies on other vertebrates (Lacy and Clark 1990; Lacy 1993; Lindenmayer et al. 1993). Lacy (1993) outlined the algorithms, structure and assumptions that underpin the program. *Emydura macquarii* was used as the study species in VORTEX to represent the freshwater turtle species of the Murray River as it has the most published data on its biology and demography, and other species are similar in their biology and demography. *Emydura macquarii* is a riverine turtle that is predominantly aquatic (Chessman 1988). The creation of additional permanent water bodies in the Murray River catchment may be beneficial for *E. macquarii* and a sympatric species, *Chelodina expansa*. Alternatively, dammed permanent water bodies may prevent dispersal and gene flow because *E. macquarii* and *C. expansa* rarely migrate terrestrially. Hence, distinct metapopulations may exist between dams, as well as in billabongs that are disconnected from the river except during major floods. Data for VORTEX were harvested from published studies (Table. 2.1; Spencer and Thompson 2005 and Spencer et al. 2017), although limited data exists for some life history attributes, thus values for these attributes were estimated based on average values for freshwater turtles worldwide.

Table. 2.1. Base parameters used for modelling extinction probability of *E. macquarii* in VORTEX

Vortex Parameter	Value	Reference
Inbreeding depression	No	

Breeding system	Polygynous	
Females in breeding pool	100%	R.J. Spencer unpublished data
% females breeding	100%	R.J. Spencer unpublished data
Age of first reproduction (**♂ /♀)**	6/10	Kennet et al. 2009
Maximum age of reproduction	70	Kennet et al. 2009
Maximum number of broods per year	1	Cann 1998
Clutch size	10-20 (average 15)	Cann 1998
Offspring sex ratio	50/50	R.J. Spencer unpublished data
Mortality 0-1 years (forest) 0-1 years (non-forest) 1-2 years 2-3 years 3-4 years 4-5 years 5-6 years 6-7 years **(♀)** 7-8 years **(♀)** 8-9 years **(♀)** Adult	 75% (± 25%) 95% (± 10%) 50% (± 10%) 40% (± 10%) 30% (± 10%) 20% (± 10%) 15% (± 10%) 10% (± 10%) 5% (± 5%) 5% (± 5%) 5% (± 5%)	Spencer 2002; Spencer and Thompson 2005
Initial population size Lagoon Lake Backwater Pond Tributary Anabranch	 300 400 200 100 300 300	J. Van Dyke unpublished data
Survival of dispersers	95%	
Dispersal rate Lagoon Lake Backwater Pond Tributary Anabranch	 1% 0.5% 1% 0.5% 1% 1%	
Carrying capacity (K)	2000	
Iterations	200	
Years modelled	200	

In the models, initial population sizes were based on relative water surface area of each habitat type, and assumed uniform density at all sites. Ponds were smallest, and populations were set at 100 turtles. Backwaters were double the size of ponds (N=200) and lagoons, anabranches and tributaries were set slightly larger than backwaters (N=300). Lakes had the largest populations of 400 turtles. Dispersal rates of Murray River turtles are not known, but in freshwater turtles, movements depend strongly on the access to water in their habitat (Hall & Steidl, 2007; Stone, 2001). In our models, waterbodies connected to the river (i.e. lagoons, backwaters, anabranches and tributaries) had an annual dispersal of 1% per year, while unconnected waterbodies (lakes and ponds) had a dispersal of 0.5% per year. Dispersal rates between wetlands separated by a dam were set at 0.1% per year.

Scenarios used in VORTEX

I created two scenarios based on high and low densities of available wetland habitats between dams on the Murray River. A high-density scenario was created that contained 21 available habitats (3 connected lagoons, lakes or backwaters and 18 disconnected ponds). A low-density scenario was created that contained seven habitats (2 connected lagoons, lakes, or backwaters and 5 disconnected ponds; Fig. 2.2). The default nest predation rate for each population was 95±5%, which is based on nest predation rates of >90% throughout the River (Thompson 1983, Spencer 2002). In both scenarios, the potential for forests to reduce nest predation rates was modelled via "relief wetlands", in which annual nest predation rates averaged 75%±25%. The percentage of relief wetlands within each modelled landscape was set as 10, 30, 50, 70, or 100%. In real-world terms, this means forests were modelled as covering 10, 30, 50, 70, or 100% of the terrestrial habitat surrounding the wetlands in each model.

I next modelled the impacts of the creation of dams within these systems. A dam was placed in the middle of each system to restrict up- and downstream movement but maintain lateral movement across the floodplain. After the dam was in place, I randomly allocated relief wetlands throughout the system. I also created an additional scenario where the dam separated relief wetlands from the high nest predation wetlands, a situation that is likely to occur on the Murray River because of land clearing for agriculture and development. One thousand iterations were run for each model, and I estimated stochastic population growth rates (r) and probabilities of extinction for each population (PE) for each scenario.

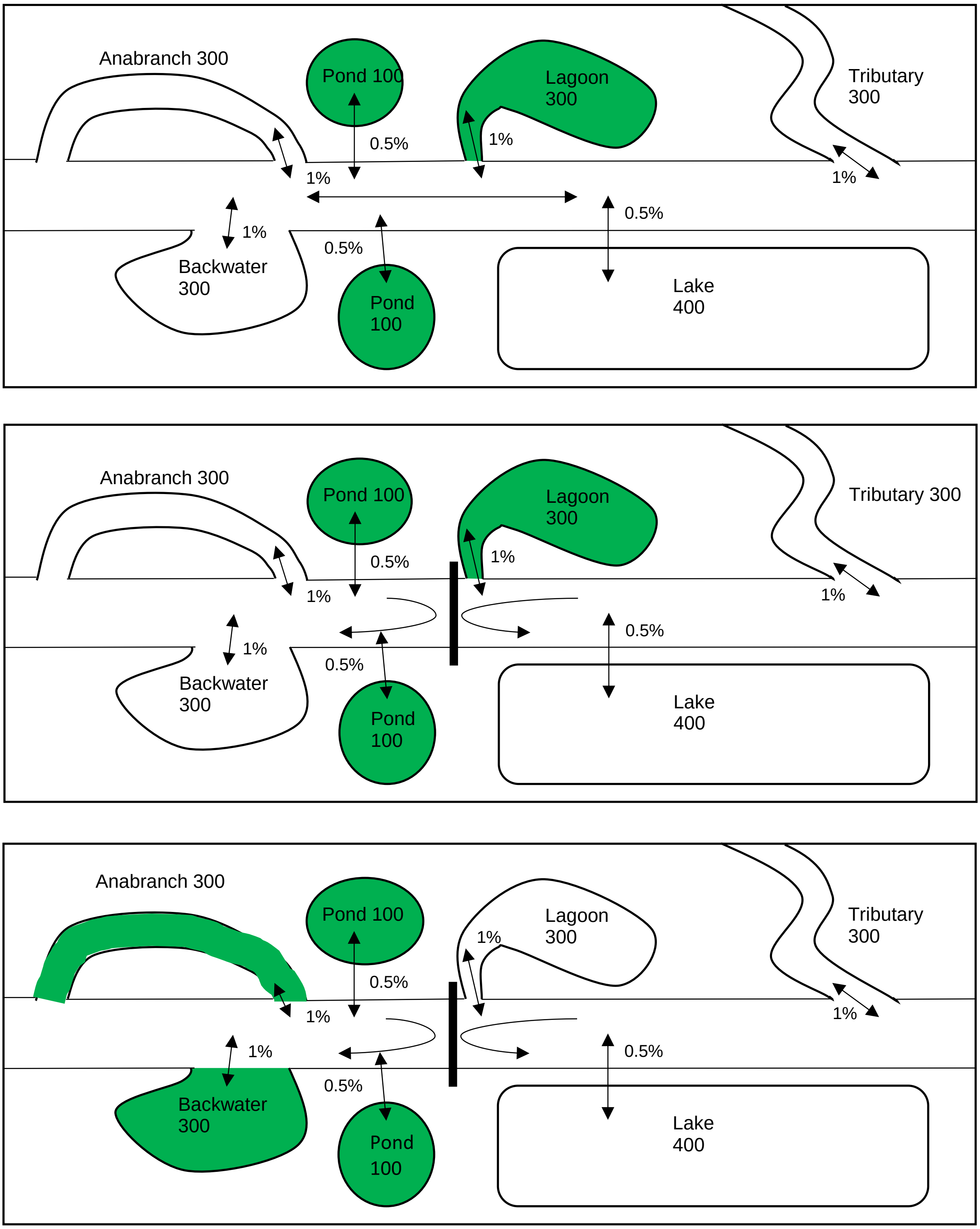


Fig. 2.2. Model scenarios based on our low wetland density models. (a) relief wetlands are shown in green and were randomly located throughout the system. (b) similar to (a), however, a dam was located in the system and (c) relief wetlands located on one side of the dam. The proportion of relief wetlands were varied in our models. Our high wetland density models were conceptually similar, but contained 21 wetlands instead of 7

2.4 Results

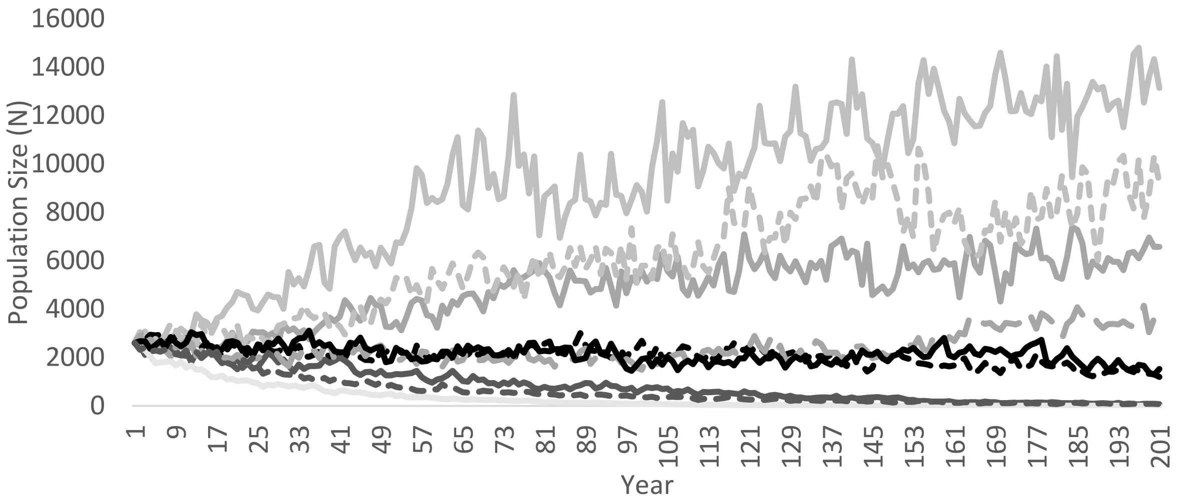
More wetland habitats are available between dams in the upper Murray River than in downstream reaches (Table 2.2), although it is prime agricultural land and the number of forested habitats are low and primarily restricted to large National Parks or State Forests. Downstream of Mildura through to Lock 1 (ie. South Australia), the number of available habitats between locks is extremely low; often less than 20 water bodies occur. More forested habitat occurs in these areas though (Table 2.2).

Table 2.2. The number of habitats up- and down-stream of each impoundment, as well as the percentage of habitats considered "forest".

	Upstream		Downstream	
Forest	Number of Habitats	Impoundment	Number of Habitats	Forest
12%	428	Yarrawonga	375	40%
40%	375	Torrumbarry	290	40%
40%	290	Euston	152	54%
54%	152	Mildura	29	22%
22%	29	Lock 10	24	73%
73%	24	Lock 9	16	100%
100%	16	Lock 8	23	100%
100%	23	Lock 7	49	58%
58%	49	Lock 6	38	67%
67%	38	Lock 5	29	64%
64%	29	Lock 4	58	59%
59%	58	Lock 3	56	10%
10%	56	Lock 2	54	58%
58%	54	Lock 1	80	7%
7%	80	Lake Alexandrina	62	7%

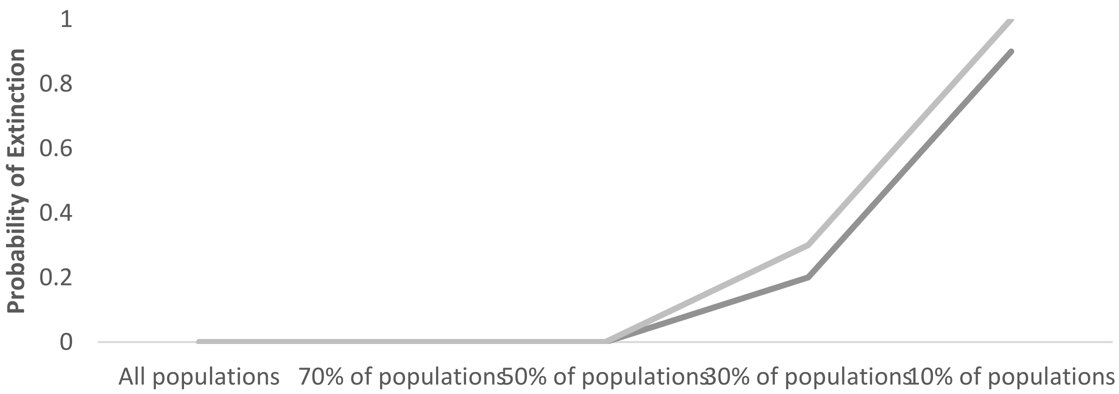
In our modelled areas with high wetland density, turtle population sizes increase both with and without dams when more than 70% of wetlands experience nest predation relief (Fig. 2.3a), although dams do reduce population growth compared to non-dam scenarios (Fig. 2.3b).

Once the number of relief wetlands reduces to below 50% in an area, populations decline. Populations rapidly become extinct when relief wetlands reduce to 10% frequency (Fig. 2.3c).



(a)

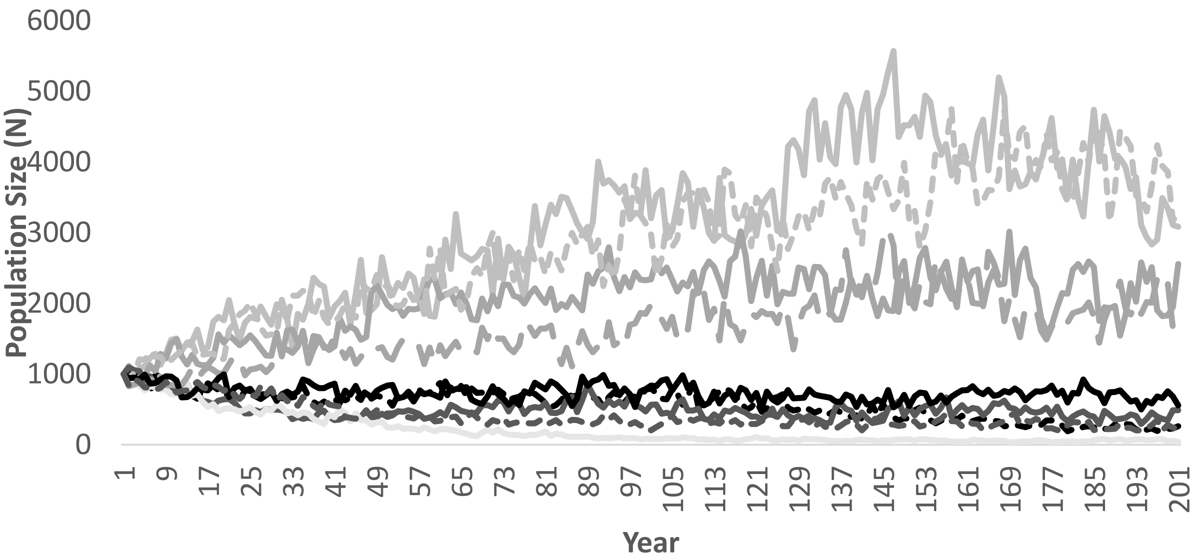
(b)



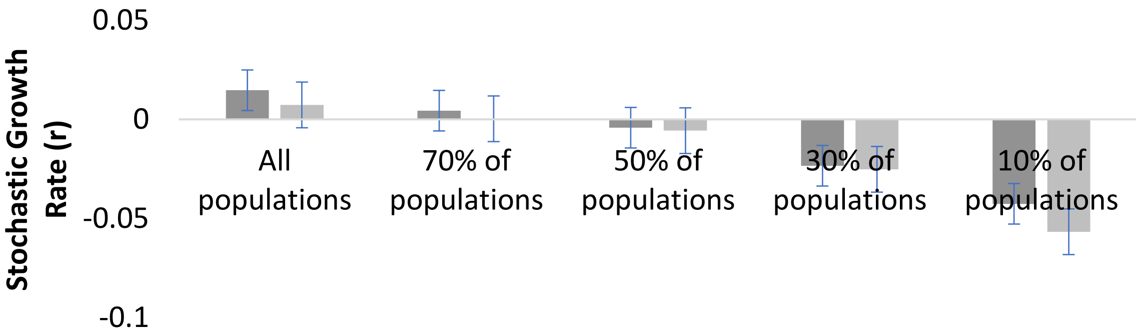
(c)

Fig. 2.3. (a) Population size over two hundred years in high wetland density systems where variable recruitment occurs in 70% (top grey lines), 50% (middle grey lines), 30% (black lines) and 10% (bottom grey line) of wetlands with (dashed line) and without a dam (solid line) allocated to the system. (b) Stochastic growth rates (r) in our high wetland density models with (light grey) and without (dark grey) a dam with various levels of relief wetlands. (c) Probability of extinction in our high wetland density models with (light grey) and without (dark grey) a dam with various levels of relief wetlands.

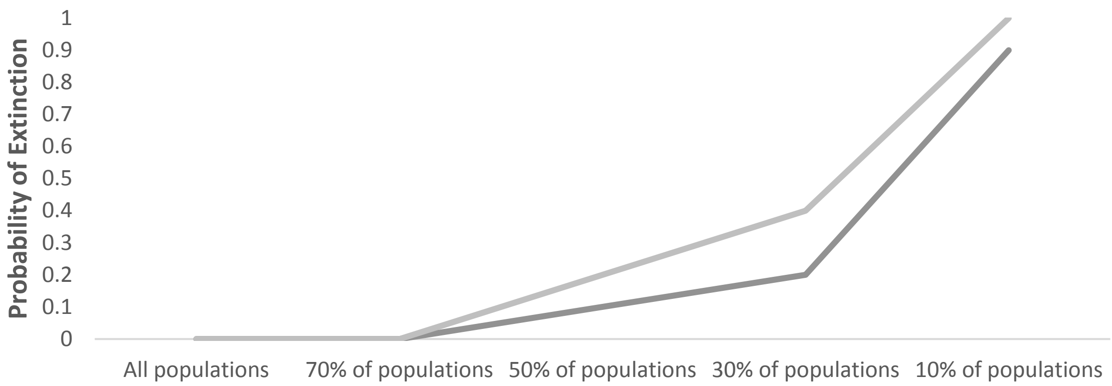
Similar patterns are observed in scenarios with low wetland density (Fig. 2.4a-c), however the risk of extinction increases more rapidly with fewer relief wetlands compared to scenarios with high wetland density (Fig. 2.4a). Adding a dam reduces population growth and increases the risk of extinction (Fig. 2.4c).



(a)



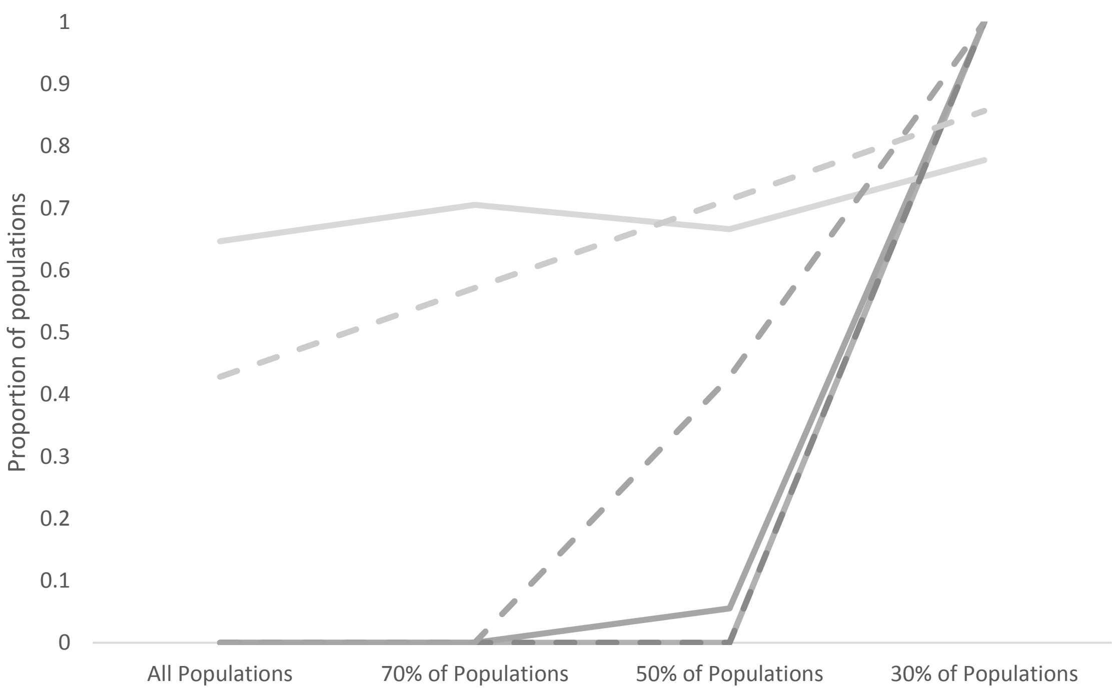
(b)



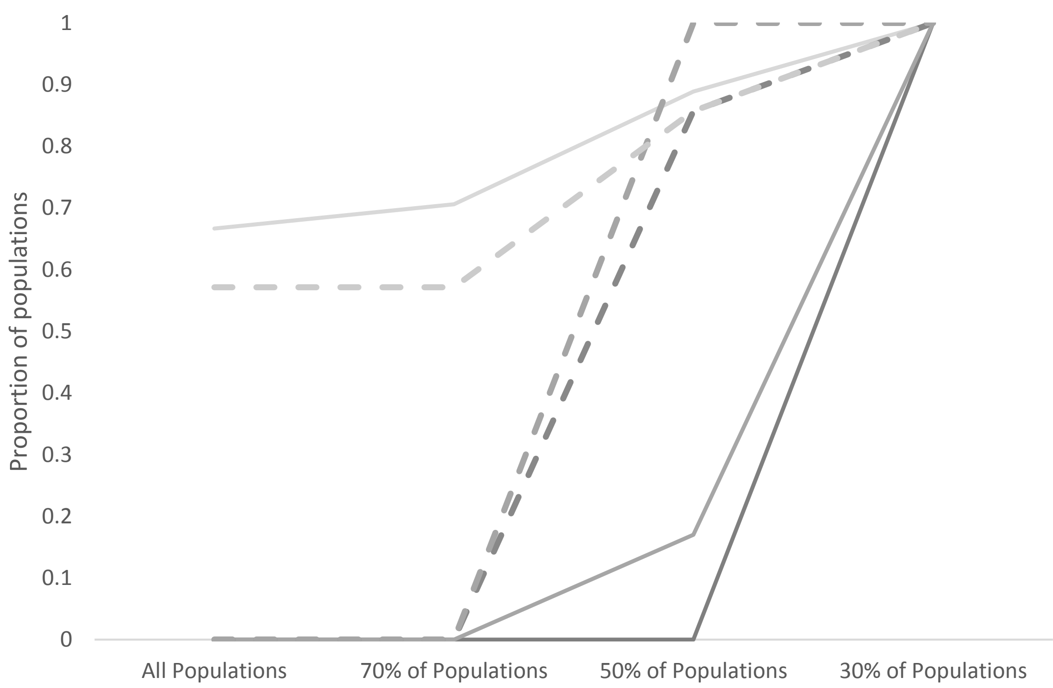
(c)

Fig. 2.4. (a) Population size over two hundred years in low wetland density models where variable recruitment occurs in 70% (top grey lines), 50% (middle grey lines), 30% (black lines) and 10% (bottom grey line) of wetlands with (dashed line) and without a dam (solid line) allocated to the system. (b) Stochastic growth rates (r) in our low wetland density models with (light grey) and without (dark grey) a dam with various levels of relief wetlands. (c) Probability of extinction in our low wetland density models with (light grey) and without (dark grey) a dam with various levels of relief wetlands.

When relief wetlands are only present on one side of a dam, more populations are driven to extinction in at least one model iteration compared to populations with no dams or where relief wetlands are randomly spread on both sides of a dam, even when relief wetlands are prevalent (Fig. 2.5a and Fig. 2.5b). The proportion of populations with high probabilities of extinction (i.e., are driven to extinction in more than 50% of our model iterations) dramatically increases in our low wetland density scenario once relief wetlands decline to less than 50% (Fig. 2.5b).


1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Proportion of populations
All Populations
70% of Populations
50% of Populations
30% of Populations

(a)


1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Proportion of populations
All Populations
70% of Populations
50% of Populations
30% of Populations

(b)

Fig 5. Proportion of populations displaying (a) any risk of extinction (>0 probabilities) or (b) high probabilities of extinction (>50% probabilities) from models without dams (dark grey), with dams that have a random distribution of relief wetlands (grey), and with dams that have clumped distributions of relief wetlands (light grey). Solid lines are our high wetland density scenarios and dashed lines are our low wetland density scenarios.

2.5 Discussion

Our models show that turtle populations of the Murray River can sustain relatively high losses of source recruitment populations (i.e., sites with low nest predation, "relief wetlands"), if they are located between dams and impoundments. Although many freshwater turtles can move terrestrially or through man-made barriers (e.g. Bennett et al. 2010), species that cannot, such as *E. macquarii*, are likely to experience local extinction events if source populations are not located regularly on the landscape between dams. Thus, the combined impact of reduced recruitment rates and population fragmentation has significant management implications on a landscape scale. Although our models do not account for the potential effects of population isolation on genetic drift or inbreeding, only weak levels of connectivity are likely sufficient to prevent them (Lacy, 1987).

The results of our models likely also apply to both *Chelodina* species in the Murray River since they face similar threats to *E. macquarii*. Our models show that source populations must be located in areas of high connectivity between impoundments to prevent extinction. The stable populations of *C. expansa* (Chessman 2011), combined with the absence of genetic divergence (Baggiano 2012) throughout the Murray River suggests that this species has a relatively high frequency of source populations. In contrast, the declines of *C. longicollis* and *E. macquarii* (Chessman 2011) suggest that there are not enough source populations for these species on a landscape scale. Notably, *C. expansa* is a solitary nester, while *C. longicollis* and *E. macquarii* nest in relatively high densities, which may explain this species difference (Spencer et al. 2016).

Obligate aquatic species are unable to overcome within-network barriers such as dams and locks, or undertake out-of-network movements (Grant et al. 2007), which elevates their risk if source populations are infrequent throughout fragmented landscapes. While terrestrial movement may mitigate the effects of aquatic fragmentation, it also carries significant risk. *Chelodina longicollis* has experienced declines of up to 91% (Chessman 2011). They often migrate terrestrially, and a recent study demonstrated that deaths due to road mortality and fox predation are significant and widespread (Spencer et al. 2017). As little as 2% harvest of the adult population can drive populations to extinction (Spencer et al 2017).

Our habitat survey shows that South Australia has few potential wetland habitats, and may be of most concern for turtle extinction. In our models, the number of available habitats between dams (Table 2.2), combined with high nest predation rates increased the risk of extinction significantly. Water quality has also affected turtles in South Australia. In early 2008, infestations of the Australian tubeworm (*Ficopomatus enigmaticus*) were reported in both *C. longicollis*, and *E. macquarii*, near the mouth of the Murray River in South Australia. Tubeworm infestation occurs due to high salinity in normally freshwater systems, and reported cases spread upstream until 2011 (Bower et al. 2012). The worms form dense layers of calcareous tubes on turtle shells, and cause them to drown. They may have killed large numbers of turtles in this region, but the exact number is not known. With little recruitment in the region and limited dispersal opportunities due to the number of dams, populations of Murray River turtles are now locally extinct in some areas of South Australia (Van Dyke et al. *in review*).

In some parts of the Murray-Darling Basin (MDB), dams may have benefited turtles by buffering the effects of droughts. Dams may allow pockets of wetland habitat to remain during severe droughts, where some turtles can survive. The restoration of large seasonal flows would then re-enable connectivity and recolonisation (Roe and Georges, 2009, Baggiano 2012). The MDB turtles, possibly with the exception of *C. longicollis*, display a 'networker' type of dispersal mode, where individuals are restricted to waterholes in times of no flow but are able to disperse rapidly among waterholes within the river network when hydrological connectivity is reinstated (Sheldon *et al.*, 2010, Baggiano 2012). Thus, although dams are detrimental in our model because they inhibit dispersal, they could also act as temporary refuges during severe droughts.

Our models show that management of turtles in the MDB should centre on increasing the number of source populations, increasing connectivity between populations, and minimising risks for species moving terrestrially. Creating harvest populations for translocation may be one solution for artificially increasing recruitment and connectivity. *Ex situ* conservation tools, such as captive breeding for reintroduction, are considered last resort to help recover threatened or endangered species. However, they may also provide alternative strategies where reducing threats directly is difficult or ineffective. Spencer *et al.* (2017) demonstrated that headstarting from harvest populations eliminated all risks of extinction, while also maintaining population growth in *Chelodina longicollis*. One harvest population could supply enough hatchlings to supplement 25 other similar sized populations at an annual rate to eliminate the risk of extinction (Spencer *et al.* 2017). Given the absence of genetic divergence (Baggiano 2012) throughout the Murray River, there is likely to be minimal loss of genetic diversity even if source populations are reduced by 75%. Thus, the concern of most *ex situ* conservation tools regarding loss of genetic diversity, is minimal in this system. However, regularly sourcing hatchlings for translocation from different source populations in a region would help prevent loss of genetic diversity and genetic divergence between populations. In this way, translocation would artificially create regular connectivity between populations in the absence of minor and moderate floods.

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3. Cost to benefit comparison of fox baiting and headstarting as freshwater turtle management methods

3.1 Abstract

The introduced red fox (*vulpes vulpes*) is a cause of freshwater turtle declines in the Murray River, Australia. The cost benefit ratios for fox control methods are unknown. 1080 baiting is most commonly used for fox control, and may cause up to 75% reductions in fox numbers in areas where it is used. Headstarting may also increase population sizes in species that experience high juvenile mortality. The cost of 1080 baiting is estimated as $23.84 AUD per bait, and the cost of headstarting as $22 AUD per hatchling. This study used VORTEX to model impacts of 1080 baiting and headstarting on turtle populations in areas of the Murray River containing high and low numbers of waterbodies. It evaluated the quantity of hatchlings and baits required for each management technique to be successful, and the overall cost of each management method. It also modelled the effects of forest cover on the cost of each management method, because nest predation rates are sometimes lower in forests. Headstarting had a significantly lower cost-to-benefit ratio. Headstarting was able to prevent extinction at a cost of $0.22 AUD per hectare, while 1080 baiting cost $7.28 to $8.73 AUD per hectare to achieve the same goal. These results suggest that headstarting may be a cost-effective method for preventing turtle nest destruction by invasive predators.

3.2 Introduction

The world's freshwater turtles are declining due to a lethal combination of natural and human induced factors. Habitat loss, climate change, introduced predators, disease, and overharvest are reducing wild populations at an unprecedented rate (Spencer 2000; Turtle Conservation Coalition 2011), with 72% of the known species of freshwater turtle in danger of extinction (Turtle Conservation Coalition 2011). On average, freshwater turtles take 5-10 years to reach maturity (Shine and Iverson, 1995; Georges et al. 2003). Once sexually mature, each adult female can produce one (or occasionally more) clutch of up to 20 eggs every year until death (Buhlmann et al. 2008). High nest predation, combined with the removal of small numbers of adult females from the population, however, can cause rapid population declines (Turtle Conservation Coalition 2011).

In Australia, one of the primary threats to freshwater turtles is the red fox (*Vulpes vulpes*; Spencer 2000; Spencer and Thompson 2005). Introduced in the mid 1800s, the fox destroys nests and reduces the number of surviving hatchlings, which reduces the future breeding population (Thompson 1983). Foxes also prey upon nesting females exposed while nesting, reducing breeding potential in future years (Spencer and Thompson 2005). Turtles inhabiting the Murray River are particularly susceptible due to the growing populations of red fox (Thompson 1983). In the 1980’s there was an estimated biomass of freshwater turtles in the Murray River of up to 100,000 tonnes, however by 2011 these populations had suffered a decline of up to 91% (Chessman 2011; Thompson 1983). The Murray River is home to three species of freshwater turtle: the Murray River short neck turtle (*Emydura macquarii*), eastern long neck turtle (*Chelodina longicollis*), and broad shelled turtle (*Chelodina expansa*). *Chelodina longicollis* and *E. macquarii* are the species of most concern, with *C. longicollis* populations suffering the most dramatic population decline of 91%, while *E. macquarii* populations have followed with a decline of 69%. *Chelodina expansa* populations may have also suffered a decline, but their low initial density makes declines difficult to detect (Chessman 2011).

Poison baiting is a popular method of fox control (Kinnear et al. 1988). Baiting is generally carried out using 1080 poison (sodium fluoroacetate) inserted into pellets or pieces of fresh or dried meat (Thomson et al. 2000). 1080 poison can be safe for humans and non-target animals while effectively reducing fox and other exotic pest numbers (Thomson et al. 2000). Fox baiting is a common method of fox control Australia, but its success at improving native prey populations is questionable (Spencer et al. 2017). In New South Wales, 1080 baiting campaigns successfully reduced rabbit numbers by 90% and fox numbers by 75%, with an increase in population sizes of and no negative effects on non-target birds and mammals in the area (Mcilroy and Gifford 1991). Populations of both pest species began recovering soon after the baiting campaign concluded, indicating a need for continued control measures (Mcilroy and Gifford 1991). Few studies follow the fate of prey species once baiting has concluded, and many baiting studies focus only on the reduction in predator populations while overlooking effects on prey species (Kopf et al. 2017).

Methods have also been investigated to protect turtle nests themselves, including covering nests with mesh grids (Yerlia 1997) and headstarting. Headstarting is the process of artificially breeding, incubating, and hatching eggs of a species before release into the wild (Heppell et al. 1996; Mitrus 2005). Headstarting eliminates the potential for eggs to be destroyed in the nest and reduces predation on nesting females, therefore giving these populations a ‘head-start’ in life (Heppell et al. 1996; Mitrus 2005). Headstarting has most commonly been implemented for marine turtles, such as the Kemps Ridley sea turtle

(*Lepidochelys kempii*; Shaver and Wibbels 2007) and the green sea turtle (*Chelonia mydas*; Bell et al. 2005). Headstarting has also been implemented for reptile and amphibian species such as the West Indian iguana (*Cyclura* sp. Alberts et al. 2004), the green iguana (*Iguana iguana*; Escobar et al. 2010), the plains garter snake (*Thamnophis radix*; King and Stanford 2006) and the Caribbean rock iguana (*Cyclura* sp.; Alberts 2007). Headstarting has also been implemented for the Plymouth red-bellied turtle (*Pseudemys rubriventris bangsi*; Haskell et al. 1996). Headstarting the European freshwater pond turtle (*Emys orbicularis*) produced a survival rate of 42% compared to only 7.5% for wild hatchlings (Mitrus 2005). Similarly, headstarting the yellow spotted river turtle (*Podocnemis unifilis*) and the arrau turtle (*Podocnemis expansa*) increased the populations of both species (Townsend et al. 2005). Headstarting has also been criticized because its effectiveness is low for species suffering from high adult mortality (Congdon et al. 1993; Heppell et al. 1996). One potential benefit for headstarting turtles in the Murray River however is that it may be highly effective in systems where mortality is most prevalent in egg/hatchling stages (Heppell et al. 1996; Spencer et al. 2017).

One of the main factors in determining whether a management method is viable is the cost-to-benefit ratio, i.e. how much the technique will cost to be successful. This study aims to determine whether 1080 baiting or headstarting would be more cost effective for preventing turtle extinctions in the Murray River. I modelled impacts of 1080 baiting and headstarting on turtle populations in areas of the Murray River containing high and low numbers of waterbodies. I evaluated the quantity of hatchlings and baits required for each management technique to be successful, and the overall cost of each management method. I also modelled the effects of forest cover on the cost of each management method, because nest predation rates are sometimes lower in forests.

3.3 Materials and Methods

The Murray River is Australia’s longest river. The river is 2,508 long, originates near Mount Kosciuszko in the Australian Alps, forms the border between New South Wales and Victoria, flows northwest into South Australia, and reaches the ocean near Lake Alexandrina (Sinclair 2001; Young 2001). The river flows through several lakes that fluctuate in salinity, including the Coorong and Lake Alexandrina, before emptying through the Murray Mouth into the Indian Ocean (Sinclair 2001; Young 2001). In chapter 2, I quantified the number of wetlands available to turtles throughout the length of the river in 10km x 10km (10,000ha) segments using google maps (see Chapter 2).

I used VORTEX (Version 10.0) to model extinction probability, and how they changed after the application of a range of management strategies, in Murray River freshwater turtles, following previous studies on other vertebrates (Lacy and Clark 1990; Lacy 1993; Lindenmayer et al. 1993). Lacy (1993) outlined the algorithms, structure and assumptions that underpin the program. *Emydura macquarii* was used as the study species in VORTEX to represent the freshwater turtle species of the Murray River as it has the most published data on its biology and demography, and other species of turtles are similar in their biology and demography. Data input into VORTEX were harvested from published studies (Spencer and Thompson 2005), and unpublished data by R. Spencer (2017; Table. 3.1). Limited data exist for some attributes, therefore values for these attributes were estimated based on average values for freshwater turtles worldwide (Shine and Iverson 1995).

Table. 3.1. Base parameters used for modelling extinction probability of *E. macquarii* in VORTEX

Vortex Parameter	Value	Reference
Inbreeding depression	No	
Breeding system	Polygynous	
Females in breeding pool	100%	R.J. Spencer unpublished data
% females breeding	100%	R.J. Spencer unpublished data
Age of first reproduction **(♂ /♀)**	6/10	Kennet et al. 2009
Maximum age of reproduction	70	Kennet et al. 2009
Maximum number of broods per year	1	Cann 1998
Clutch size	10-20 (average 15)	Cann 1998
Offspring sex ratio	50/50	R.J. Spencer unpublished data
Mortality 0-1 years (forest) 0-1 years (non-forest) 1-2 years 2-3 years 3-4 years 4-5 years 5-6 years 6-7 years **(♀)** 7-8 years **(♀)** 8-9 years **(♀)** Adult	 75% (± 25%) 95% (± 10%) 50% (± 10%) 40% (± 10%) 30% (± 10%) 20% (± 10%) 15% (± 10%) 10% (± 10%) 5% (± 5%) 5% (± 5%) 5% (± 5%)	Spencer 2002; Spencer and Thompson 2005

Initial population size Lagoon Lake Backwater Pond Tributary Anabranch	 300 400 200 100 300 300	J. Van Dyke unpublished data
Survival of dispersers	95%	
Dispersal rate Lagoon Lake Backwater Pond Tributary Anabranch	 1% 0.5% 1% 0.5% 1% 1%	
Carrying capacity (K)	2000	
Iterations	200	
Years modelled	200	

I used the high population density models created in Chapter 2 as the basis for modelling the impact of different management strategies. In essence, 21 populations were created and nest predation rates at each population was set at high levels (95%± 5%). A range of management scenarios were created. These included incrementally reducing nest predation rates in the connected and disconnected populations, as well as supplementing these populations with incrementally increasing numbers of headstarted hatchling turtles. For these scenarios headstarted turtles were released at the time of hatching (ie not raised in captivity).

Costs of fox baiting programs and their effectiveness were evaluated from two programs being conducted in Victoria. North Central Catchment Management Authority have been conducting standard baiting campaigns in Gunbower State Forest (latitude -35.805 longitude 144.266) since 2014 to manage fox predation on turtle populations. Baiting has been conducted up to twice a year and nest predation rates were evaluated before and after baiting. A larger scale baiting campaign has been conducted by Victoria DEWLP and the Goulburn-Broken Catchment Management Authority in Barmah National Park (latitude -35.977, longitude 144.993). Management agencies were asked to provide information on the scale of the program (hectares and time), costs of the baiting program (costs per bait and costs per hectare), as well as, evaluation of success of the program (number of baits taken, reduction in fox activity and reduction in nest predation rates).

For headstarting, I conducted a literature review and a surveyed fellow researchers via the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group (TFTSG) to determine the costs per hatchling released. The questions asked in the headstarting survey included:

1. location of program
2. species
3. estimated setup costs
4. size of breeding population
5. annual on-going costs
6. age at release
7. number of turtles released per year
8. hectares of water serviced by the release
9. how long your program has been running
10. what would you have done differently or other qualitative info that is worth sharing.

3.4 Results

The responses collected from this survey, combined with results obtained through a literature review were averaged to obtain a cost per hatchling/bait (Table. 3.2; Table. 3.3). Baiting was determined to cost $23.84 AUD per bait, with each bait reducing nest predation by 0.063%, while the costs of each hatchling was $22 AUD. The results from Woody (1991) and Ogle were not used as they did not take into consideration full set-up and on-going costs and had limited information available.

Table. 3.2. Cost break down of baiting for *E. macquarii*

Cost per bait	Area (ha)	Baits used	Reduction in nest predation	Cost/annum	Cost/ha	Reference
$23.55	4908	1274	80%	$30,000	$6.11	Unpublished Data, K. Howard (DEWLP)
$24.12	1200	390	None observed	$9408	$7.84	Unpublished data A. Martins (NCCMA)

Table. 3.3. Cost break down of headstarting for *E. macquarii*

Cost per hatchling	Area (ha)	hatchlings supplemented	Cost/annum	Cost/ha	Reference
$22	10000	1000	$22,000	$2.20	Unpublished data, R. Burke
$0.81	Unknown	2000	$1,620	Unknown	Woody (1991)
$53.97	45	176	$9,500	$211	Unpublished data, M. Ogle

Fox management through baiting was successful when the area covered was close to 5000 ha, where an 80% reduction in nest predation rates were observed (Table 3.2). Fox baiting over smaller areas does not appear effective (Table 3.2).

An 80% reduction in nest predation significantly reduces declines and eliminates the risk of extinction. In my models, an 80% reduction in nest predation rates only around the three river connected populations can eliminate extinction risk from all other 18 populations that are relatively isolated in the landscape (Fig. 3.1).

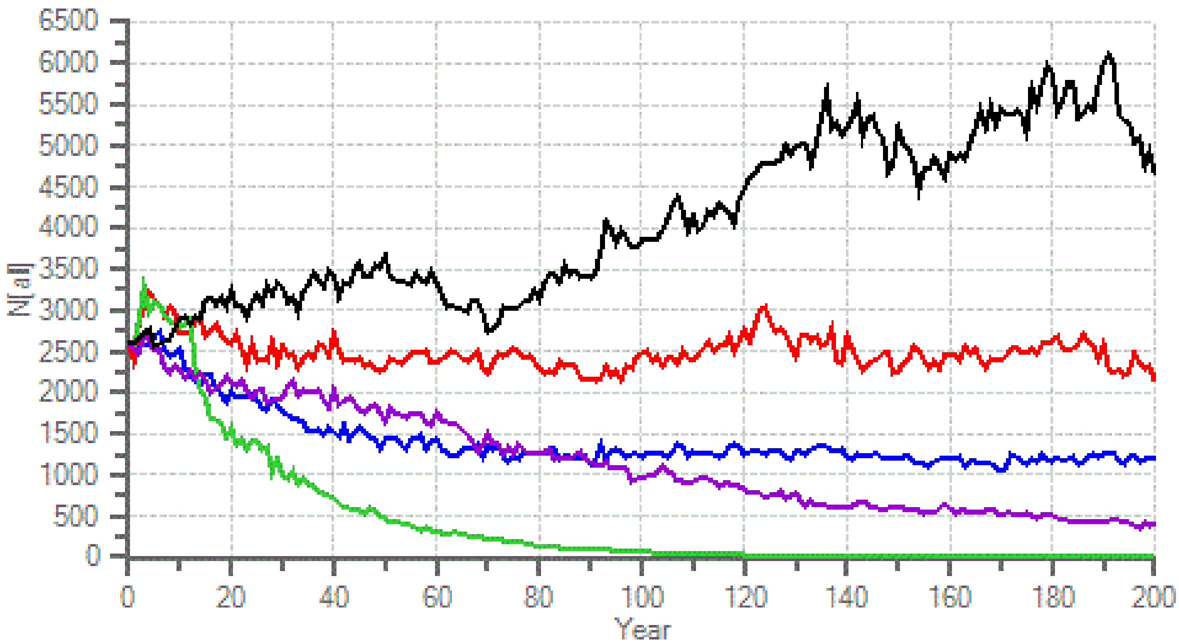

Fig. 3.1. Population size (N) over 200 years when (1) a 80% reduction in nest predation rates are applied to connected populations (black line) (2) Headstarting of 200 turtles per annum (600 total) are released into each connected population (red line) (3) Headstarting of 100 turtles per annum (300 total) are released into each connected population (Blue Line) (4) a 50% reduction in nest predation rates are applied to connected populations (purple line) and (5) 600 headstarted hatchlings are evenly scattered throughout 18 isolated ponds in the landscape (green line).

Headstarting 600 hatchlings into connected populations maintains population stability and eliminates extinction risk (Fig. 3.1). The same number of hatchlings scattered evenly throughout the system is not as effective (Fig. 3.1). The release of 600 hatchlings effectively represents 23% of the initial population size of the system or effectively supplementing populations with eggs from an extra thirty turtles per year.

Based on my google maps analyses, the 21 habitats/populations used in these models covers an area of 3,000- 210,000 ha along the River (Fig. 3.2). Based on my estimated per-hectare cost of successful baiting effort (~$6 per ha; Table 3.2), effective baiting campaigns may between cost $30,000 (over 5000 ha) and $1,260,000 (210,000 ha) per year. In comparison, our models show that headstarting only 600 turtles would maintain the same populations. 600 hatchlings would cost between $13,200 and $32,400 to cover the same area based on the costs per hatchling in Table 3.3.

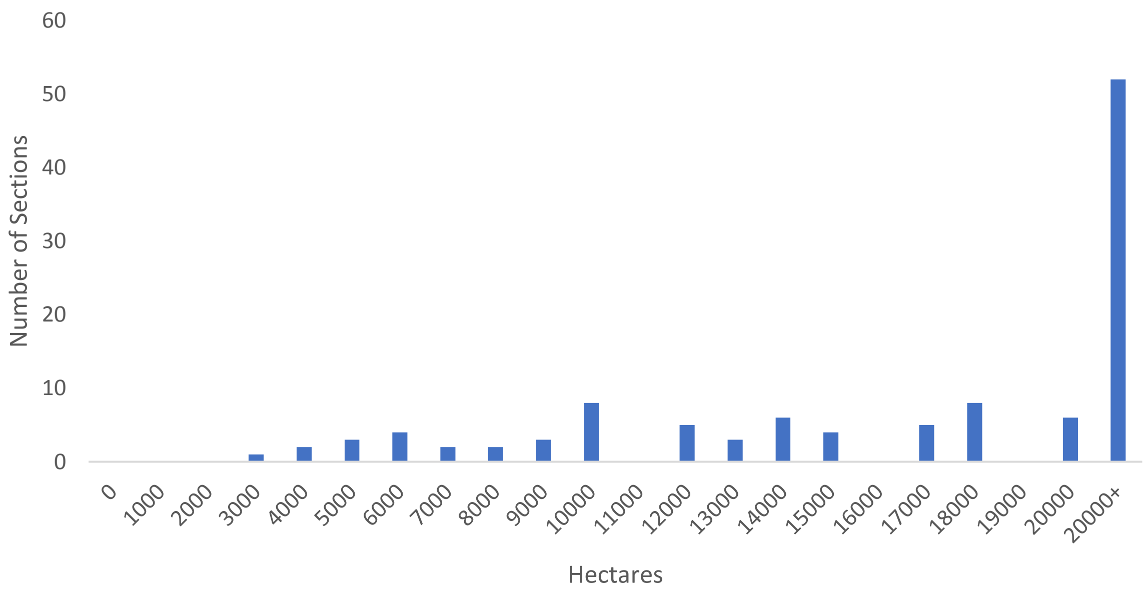

Fig. 3.2. A histogram representing the area that our 21 model populations covers in sections of the River.

3.5 Discussion

There is a dire need for financially viable methods of freshwater turtle management throughout the world, and the Murray River turtle community is a prominent example (Mitrus 2005; Spencer and Thompson 2005). My models show that baiting and headstarting could each reduce turtle extinction risks, but headstarting, in most circumstances, is far more economically viable. Based on my estimated per-hectare cost of successful baiting effort, effective baiting campaigns may cost between $30,000 (over 5000 ha) and $1,260,000 (210,000 ha) per year. In comparison, our models show that headstarting only 600 turtles, or 23% of the initial population, would maintain populations. 600 hatchlings would cost between $13,200 and $32,400 to cover the same area based on the costs per hatchling.

These results can be used to improve conservation outcomes for turtles, as they provide evidence that headstarting may prevent turtle extinctions more effectively than 1080 baiting. Currently, 1080 baiting is one of the most common methods to control foxes in Australia (Greentree 2000; Saunders and McLeod 2007). Despite its widespread use, few conservation benefits have been observed (Walsh 2012; Spencer et al. 2017). In Gunbower Forest, Victoria, a baiting and shooting campaign was run covering an area of 1200ha (Unpublished data A. Martins). Shooting cost $9.33/ha, equating to $11,196, while baiting cost $7.84/ha, or $9,408 in total. Despite the cost of baiting being more than three times than my estimate, and shooting being even more expensive, no changes in fox or turtle numbers were observed (Unpublished data A. Martins). Similarly, 23 years of fox baiting had little benefit for malleefowl in South Australia (Walsh et al. 2012). There was no significant decrease in fox numbers observed, and malleefowl population growth did not occur (Walsh et al. 2012).

Furthermore, my study demonstrates the inconsistency in the impact of baiting. As listed in table 3.2, NCCMA baiting cost more per bait and per hectare and used more baits per hectare, but had less impact than K. Howard's study, likely because it took place over a smaller region. This indicates that baiting intensity alone is insufficient if it is only conducted in a relatively small area, and both high-intensity baiting (many baits per ha) and a large area of baiting are required to have an impact. These two requirements require a much higher cost than conducting either a low-intensity/large area effort, or a high-intensity/small area effort.

Unlike baiting, headstarting has potential genetic consequences. Baiting allows many females to continue producing hatchlings over time, whereas headstarted hatchlings are produced by the small pool of males and females selected for breeding. Care must be taken to ensure that genetic diversity remains adequate in the population, and that natural selection and natural evolution can continue to act on the population (Cross 2000). One way to preserve genetic diversity would be to use different parents as hatchling sources in different years at any given site. Inbreeding and genetic loss is a common problem in translocation and *ex situ* conservation (Allendorf and Ryman 1987; Cross 1999; Cross 2000). In fisheries, even one generation of artificial breeding and rearing can cause large changes in the genetic make-up of a population, including loss of genetic variability (Allendorf and Ryman 1987; Cross 1999). For translocation programs, a large breeding population is required with single pair mating favored if possible (Gharrett and Shirley 1985; Cross 2000). An equal sex ratio must also be present within the breeding population to maximize genetic diversity in breeding crosses (Cross 2000). When an inadequate breeding population is used, genetic variability is lost and genetic composition is changed due to genetic drift (Tave 1986). For example, in the extreme case that a breeding population contains only one male and one female, 25% of genetic variability is lost in each generation (Cross 2000). If a large breeding population is unobtainable, breeding adults must be replaced every few years to introduce new genetic material into the breeding pool (Allendorf and Ryman 1987). It should also be noted that headstarting success is dependent on the sites hatchlings are translocated into. Wetlands with high dispersal potential can disperse into nearby wetlands and help boost other nearby populations, however translocating into wetlands with low dispersal potential would serve little benefit to the population.

Other management techniques may also provide a solution. Fencing off important nesting grounds to exclude foxes while allowing turtles to move in and out of the area may provide a more long term management solution (Saunders et al 1995). There has been debate however about the effectiveness of management solutions such as fencing as foxes have the ability to squeeze through small holes and gaps, and have been known to dig under fences. Therefore constant maintenance and frequent monitoring would be required to ensure that the fencing remains effective (Saunders et al 1995). Another argument against fencing is the effect it has on the ecology of the enclosed area, as ‘fences can interfere with the movement of non-target animals’ (Saunders et al 1995).

In conclusion, I demonstrated evidence that headstarting programs could be a cost-effective conservation strategy for stopping the declines and extinctions of freshwater turtles in the Murray River region. Headstarting will be particularly effective for species primarily threatened by both adult mortality and invasive predators (Spencer et al. 2017). These findings can contribute to the development and evaluation of successful management techniques for

Australian freshwater turtles, as-well as similar species of freshwater turtles worldwide. However, my results only demonstrate that the concept of headstarting may be cost effective if it works. Experimental evaluations of headstarting are necessary before it is adopted as a management strategy.

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4. Conclusions and Future Work

4.1 Modelling the impacts of variable predation and water regulation on Murray River turtles

I found evidence that turtle populations of the Murray River can sustain relatively high losses of source recruitment populations, if relief populations are located between dams and impoundments. Although many freshwater turtles can move terrestrially and some are able to move directly through man-made barriers (e.g. Bennett et al. 2010), species such as *E. macquarii* are likely to experience local extirpation events if source populations are not located regularly on the landscape between dams. Reduced recruitment levels combined with demographic independence of populations has significant management implications (Chessman 2011).

Source populations are both species- and habitat-dependent. Areas with nesting grounds that provide large areas of heterogeneity for nesting habitat are likely to provide important recruitment sources for low density nesting species, like *C. expansa* (Spencer et al. 2016), however, higher density nesting species, such as *E. macquarii* and *C. longicollis*, are unlikely to experience similar relief. Thus, it is vital that source populations be located in areas of high connectivity between impoundments. Management of MDB turtles centres on increasing the number of source populations for several species, increasing connectivity between populations and minimising risks for species moving terrestrially.

4.2 Cost to benefit comparison of fox baiting and headstarting as freshwater turtle management methods

For populations to recover in the region, headstarting or translocation programs will be necessary as current fox management techniques are inadequate and costly. In my second study, I compared the cost and efficiency of headstarting and baiting, the most widely practiced fox management program, and determined that headstarting is more efficient, economically beneficial and uniformly successful than baiting. Based on my research, published research from Walsh (et al. 2012), and unpublished data from A. Martins, K. Howard, and R. Burke, it was apparent that baiting was inconsistent in its effectiveness and costlier than headstarting. Baiting was successful when it was conducted intensely over an area of close to 5000 ha. There, an 80% reduction in nest predation rates was observed. In my models, an 80% reduction in nest predation around only three populations connected to the river can eliminate extinction risk from 18 more populations that are relatively isolated in the landscape. Baiting over smaller areas however did not appear effective. These results show that while intense baiting could prevent turtle extinctions in large areas, smaller efforts would be less effective. Therefore, baiting is an inconsistent management technique.

Headstarting however showed much more uniform effectiveness. 600 hatchlings supplemented into three connected populations maintains population stability and eliminates extinction risk at all 21 sites. The same number of hatchlings scattered evenly throughout the system is not as effective. The release of 600 hatchlings effectively represents 23% of the initial population size of the system and would represent supplementing populations with eggs from an extra thirty turtles per year.

Cost-wise, an effective baiting campaign may cost between $30,000 (over 5000 ha) and $1,260,000 (over 210,000 ha) per year. In comparison, our models show that headstarting only 600 turtles would maintain the same populations, and cost between $13,200 and $32,400 to cover the same area. These dramatic price differences, and the consistency of headstarting success in all areas suggest that headstarting is an ideal management method for the Murray River's freshwater turtles

4.3 Evaluating the success of headstarting

Evaluation of the success of headstarting however may be difficult, as demonstrated in a study I conducted in Lake Bonney-Barmera, South Australia. In 2015, 2000 *E. macquarii* hatchlings that had been bred, hatched, and raised in captivity by a local turtle breeder were released into four locations along Lake Bonney-Barmera. These hatchlings were measured and toe clipped for later identification. Two years later in early 2017 a recapture study was conducted to determine the proportion of the headstarting hatchlings that had survived. Over a two-week period, recapture attempts were conducted for 3 consecutive days in each of the four release sites. Cathedral traps, dip nets, fyke nets and crab traps were used, with traps checked for turtles approximately every 9-10 hours and dip netting conducted for 1 hour at each site. Although many adult and juvenile turtles were caught during this study, none of the headstarting hatchlings released in 2015 were recaptured.

There are two possible explanations for these results. It is possible that none of the headstarted hatchlings survived, although this is unlikely as similar studies in the Murray River have produced a recapture rate of 12% (Spencer 2005). Instead, hatchlings may have dispersed further throughout the lake and adjacent Murray River than expected. During our recapture effort, an unmarked juvenile was caught, marked, and then re-caught a week later more than 5 km away. Furthermore, hatchlings may simply be difficult to capture or hatchlings may have dispersed to areas of the lake that we did not trap.

Dip nets are thought to be the most effective at capturing juvenile and hatchling turtles, particularly as they can reach the areas most inhabited by juveniles including along the edges of the river among the reeds. However, the majority of the capture effort was invested in the remaining traps and nets because we did not want to disturb baited traps while we dip-netted. Juveniles are occasionally caught in baited traps however, and it is a possibility that the bait used was not ideal for attracting juveniles. The hatchlings were all released into Lake Bonney, hence the recapture efforts were focused on Lake Bonney. However, the juveniles may have travelled out of the lake in the intervening 20 months. Recent studies in Barmah, Victoria, have shown that some turtles can travel upwards of 20 km. Although those that travelled these distances were adult males, they establish the potential for larger movements than expected (personal communication, K. Howard)

4.4 Recommendations

Management of MDB turtles centres on increasing the number of source populations for several species, increasing connectivity between populations and minimising risks for species moving terrestrially. Creating harvest populations for headstarting may be one solution for artificially increasing recruitment and connectivity. My study has shown that the most efficient headstarting management plan would be to supplement into waterbodies connected to the main channel of the Murray River (or tributaries). This would maximise hatchlingdispersal, which maximises potential to replenish the wider population.

Headstarting , while efficient and cost effective, can be hard to evaluate in terms of long-term success. Therefore, investigations need to be undertaken to determine the best ways of measuring headstarting success (i.e. best ways of trapping, best areas to trap, distance juveniles have likely travelled) before full scale headstarting programs are undertaken. Long-term monitoring of health and fecundity of headstarted turtles is necessary to ensure that headstarting programs are evaluated properly.

4.5 References

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